Risk screening assessment for ranking historic coastal landfills by pollution risk

James H. Brand[a]

[a] School of Geography
Queen Mary University of London
Mile End Road
London
E1 4NS
England
j.h.brand@qmul.ac.uk (corresponding author)

Kate L. Spencer[b]

[b] School of Geography
Queen Mary University of London
Mile End Road
London
E1 4NS
England
K.Spencer@qmul.ac.uk

Abstract

Globally there are significant numbers of historic landfills, and in England alone there are over 1200 in low-lying coastal areas. Approximately one-third of these *historic coastal landfills* are near designated ecological sites, and without intervention, 10% are expected to start eroding within 40 years. Indeed, some sites are already eroding and releasing waste, and erosion is likely to become more common with the anticipated effects of climate change. Mitigating the pollution risk from all historic coastal landfills under threat of erosion would be prohibitively expensive; consequently, it is necessary to understand which sites pose the greatest pollution risk in order to prioritise management resources. This paper proposes a new risk screening assessment that can support coastal managers in identifying which historic coastal landfills pose the greatest pollution risk at a national scale for minimal cost using existing datasets. The proposed method determines an *overall risk index* for each site by considering the risk of pollution from eroding historic coastal landfills in two stages: the first stage assesses the risk of waste being released (*waste release index*), and the second assesses the risk to various receptors (*pollution index*). The highest risk sites can then be prioritised for further investigation or remediation.

Keywords

- Estuarine and coastal management
- Risk assessment
- Historic coastal landfills
- Coastal vulnerability index
- Contaminated land management
- Flood defence management

1. Introduction

Historically it was common practise to landfill domestic, commercial and industrial waste in areas considered of limited economic value due to the risk of flooding, such as low-lying estuarine and coastal locations. For example, in England there are over 1200 historic landfills in coastal and estuarine locations that are low-lying and have a high risk of sea flooding (i.e. $\geq$0.5% annual probability) and/or erosion if not adequately defended; without intervention, 10% are anticipated to start eroding by 2055 (Brand et al., 2017). The likelihood of these historic coastal landfill sites flooding or eroding is increasing due to climate change effects such as increased sea level and more frequent extreme weather events, and this may have consequences for pollutant release. Inundation would increase leachate production, but significant dilution in open waters would minimise risk. Solid waste is usually fully contained and isolated from the marine environment by capping materials, and is often protected by flood defences (Brand et al., 2017); however, historic landfills and their defences are increasingly at risk of breaching, because inundation will increase the probability of failure through erosion, piping or excessive seepage (Defra and Environment Agency, 2007). Consequently, historic waste materials may be released into the coastal zone, and this has already occurred in some locations (Pope et al., 2011). Historic waste may include a wide range of materials that are physically harmful to ecological and public health, such as asbestos and plastics, as well as pathogens, and inorganic and organic contaminants that significantly exceed environmental quality guidelines (e.g. Brand and Spencer, in review). This poses a significant challenge to coastal managers as mitigating the risk from all historic coastal landfills is likely to be prohibitively expensive (Cooper et al., 2013; Weber et al., 2011). Therefore, it is essential to prioritise regional and national expenditure by mitigating those sites that pose the greatest pollution risk (Brand et al., 2017).

Risk is typically considered as a function of the probability of something happening and its consequences (Wamsley, 2015). There are many factors that may influence the probability that contaminated materials from historic coastal landfill sites are released, including wave exposure, the condition and design standard of any flood defences present and local coastal erosion rates (Alaska Department of Environmental Conservation, 2015). The consequences of pollution occurring are dependent on the vulnerability of the receptors (Wamsley, 2015), which can be considered as the probability that the receptors will be affected by hazards/drivers, and is often considered in terms of a dose-response relationship (Gormley et al., 2011). Therefore, the consequences of contaminated materials being released will depend upon the quantity of materials released and their contaminant loads, contaminant bioavailability and mobility, dilution by the receiving waters, and receptor sensitivity to those contaminants. In turn, the quantity of materials released will depend on many of the same factors as the probability of contaminated material release, plus the size of the landfill, i.e. quantity of waste, whether it is divided into structurally stable cells, the mechanical properties of the waste, e.g. waste cohesion, the shape of the landfill, i.e. the proportion of it adjacent to the coast, and how quickly any breach can be repaired (Alaska Department of Environmental Conservation, 2015; Cooper et al., 2013).

Combining such diverse data types into a readily understood form that indicates their combined effect can be achieved using index and indicator methods (Ramieri et al., 2011). However, many of these data are not readily available and would require impracticable levels of resources to obtain in countries with large numbers of landfills. Where detailed data are not readily available to assess risk at local, regional or national scales Rosendahl Appelquist and Balstrøm (2014) propose a three step approach to assessment, where steps 1 and 2 are used for regional or national scale assessments and step 3 is only used for local scale assessments:

Step 1. High level initial screening using remote sensing and existing data to gain a cost-efficient, relatively low accuracy overview of the risk.
Step 2. Field verification of the data used in step 1.
Step 3. Systematic and detailed field investigations for high accuracy, local level assessments of risk hot-spots identified in steps 1 and 2.

This approach has the advantage of reducing expenditure on site investigations and providing a method to prioritise resources when there are multiple sites to manage. It has the disadvantage that existing data may not highlight factors that increase risk, e.g. records may not show that a site has already started to erode.

There have been a number of attempts to apply the index and indictor approaches to the management of landfill sites both on the coast and inland, none of which have been widely adopted. These typically only consider the risk of pollution when the waste is fully encapsulated, do not consider inundation, and focus on the risk from leachates and gases (e.g. Kumar and Alappat, 2005; Okaneya et al., 2013; Sharma et al., 2008; Singh et al., 2009). Where erosion of waste as a pollutant pathway has been considered, methods are too location specific for wide application (Alaska Department of Environmental Conservation, 2015; 2009; Laner et al., 2008; Neuhold, 2013; Neuhold and Nachtnebel, 2011). Hence, a new region-specific method is required for assessing coastal landfills that can be applied in both England and physically similar temperate coastal environments.

The overall aim of this research was to develop a high-level risk screening assessment methodology, focused on the risk to the intertidal zone and tidal waters from eroding historic coastal landfills, which will support coastal managers in allocating limited resources to addressing the sites that pose the greatest pollution risk. The presented risk screening assessment approach has considered the risk of pollution from eroding historic coastal landfills in two stages: the first stage assesses the risk of waste being released, and the second assesses the risk to various receptors.

2. Developing the risk assessment

Coastal vulnerability and landfill screening assessments typically provide relative, not absolute, indications of risk by considering parameters that represent the vulnerability of receptors to specific hazards/drivers using the best available datasets (Kumar et al., 2010; Rygel et al., 2006; Sayers et al., 2003; Wamsley, 2015). Therefore, to assess the risk of waste being released (hereon referred to as the waste release index) this research has identified parameters to represent coastal drivers (e.g. wave action) and landfill vulnerability (i.e. likelihood of the landfill releasing waste). To assess the risk to receptors from eroded waste (hereon referred to as the pollution index) parameters have been identified to represent the landfill hazard (representing volumes and toxicity of waste released) and environmental vulnerability (i.e. likelihood of environmental harm from the released waste). The relationship between the sub-indices, indices and the overall risk index is illustrated in Figure 1.

The main parameters identified as important for assessing drivers of coastal landfill erosion on low-lying coasts are wave exposure, storm climate, flooding and tidal range (Laner et al., 2009; Laner et al., 2008; McLaughlin and Cooper, 2010; Neuhold, 2013; Neuhold and Nachtnebel, 2011; Rosendahl Appelquist, 2013). There is little variation in mean wind speeds and wind gust speeds around England (Met Office, 2016) and, therefore, storm climate was not considered. The main parameters for assessing the vulnerability of coasts to erosion are the coastal geomorphological type, coastal slope, sediment balance, beach width and vegetated areas (Denner et al., 2015; McLaughlin and Cooper, 2010; Rosendahl Appelquist, 2013), and landfill assessments also consider the presence/absence of flood defences and the distance from the landfill to mean high water (Alaska Department of Environmental Conservation, 2015). Here, features of the landfills were considered in place of the natural geomorphology and underlying geology.

Parameters used in the landfill risk assessments to represent the hazard depend on the overall aim of the specific method and can be summarised as quantities and types of waste parameters, and contaminant concentration parameters (e.g. Alaska Department of Environmental Conservation, 2015; Laner et al., 2009). Parameters used in the landfill risk assessments to represent the vulnerability of receptors also depend on the overall aim of the specific method and include water use, the proximity of habitats, and the presence of flora and fauna (including humans) (Cooper et al., 2013).

In the coastal vulnerability and landfill assessments, parameters are assigned relative severity scores to allow both quantitative and qualitative data to be used in the same assessment (Ramieri et al., 2011; Singh et al., 2009; Wamsley, 2015). Here, a five-point severity scale for each parameter is used with the highest values indicating the greatest hazards or vulnerabilities of receptors (Table 1) (e.g. Denner et al., 2015; Gill et al., 2014; Palmer et al., 2011). Wherever possible severity scores from existing risk assessment methods have been utilised, but new severity scoring systems are proposed where necessary. How the parameters fit into the overall assessment process is summarised in Figure 1. Some parameters are included in more than one sub-index, and therefore counted twice - once in each index, this is because they represent both the likelihood of waste being released and the rate at which waste would be released if a landfill were breached.

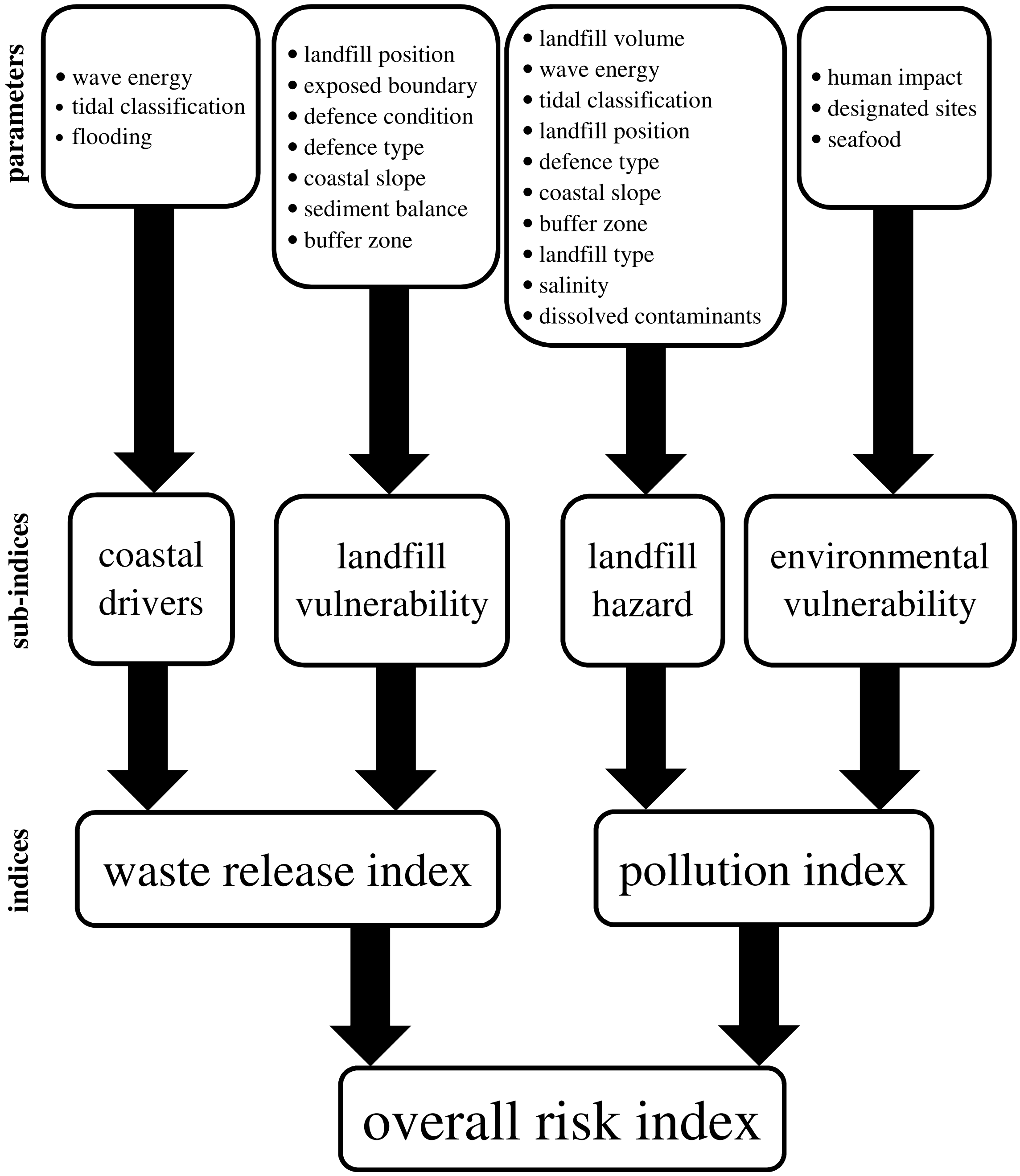

Figure 1: Flow chart showing the relationship between the parameters, sub-indices, indices and overall risk index

Table 1: Assigning severity scores to the assessment parameters for each historic coastal landfill

		Severity score				
Parameters	**Measure**	**1**	**2**	**3**	**4**	**5**
Wave energy[1]	fetch	<10 km or saltmarsh present	-	10-100 km	-	>100 km
Tidal classification	tidal range	macrotidal (>4 m)	-	mesotidal (2 to 4 m)	-	microtidal (<2 m)
Flooding	predominant RoFRS zone over landfill	predominantly outside RoFRS	very low	low	medium	high
Landfill position	landfill boundary to mean high water (m)	>50	35< to 50	20< to 35	5< to 20	≤5
Exposed boundary	length of landfill boundary facing foreshore (m)	≤500	500< to 1000	1000< to 2000	2000< to 3000	>3000
Defence condition	flood defence condition grade	1	2	3	4	5
Defence type		hard	mixed	soft	partly undefended	no defence or landfill is defence
Coastal slope[2]	distance between landfill and 20 m isobath (km)	>4	3< to 4	<2 to 3	<1 to 2	≤1
Sediment balance		accretion	-	no change	-	erosion
Buffer zone	width of saltmarsh (m)	>50	20< to 50	10< to 20	0< to 10	No saltmarsh
Landfill volume	volume (m^3)	≤500,000	500,000< to 1,000,000	1,000,000< to 1,500,000	1,500,000< to 2,000,000	>2,000,000
Landfill type	NB for mixed sites choose highest severity score of the types present	inert	MSW or commercial	industrial	special waste	liquids/sludge or unknown
Salinity		upstream of oligohaline zone	oligohaline zone	mesohaline zone	polyhaline zone	downstream of polyhaline zone
Dissolved contaminants	tidal prism volume (m^3)	>500,000,000	100,000,000< to 500,000,000	50,000,000< to 100,000,000	5,000,000< to 50,000,000	≤5,000,000
Human impact	distance to bathing water catchment (m)	>150	100< to 150	50< to 100	0< to 50	Landfill is in bathing water catchment
Designated sites	distance to designated site(s)	>250 m upstream and >1 km downstream	≤250 m upstream or 1 km downstream	≤250 m upstream or 500 m downstream	0< to <100 m	Landfill site is within designated site(s)
Seafood	distance to shellfish/mollusc site	>250 m upstream and >1 km downstream	≤250 m upstream or 1 km downstream	≤250 m upstream or 500 m downstream	0< to <100 m	Landfill site is within shellfish/mollusc site(s)

[1]categorisation after Mangor (2004) and Rosendahl Appelquist (2013) [2]adapted from Palmer et al. (2011)

2.1 Parameter datasets

2.1.1 Wave energy

Wave energy as a driver of erosion influences whether a landfill will breach and the rate at which waste would be released. Wave energy hitting the shoreline depends upon the height of waves, their orientation to shore, wave fetch, and width and vegetation of any buffer zones (Möller and Spencer, 2002; Rosendahl Appelquist, 2013). Currently, there are limited data available for most of these factors in England. In the absence of wave data, free fetch can be used to classify coasts as protected (waterbody width <10 km), moderately exposed (10 km< waterbody width <100 km) or exposed (100 km< waterbody width) (Rosendahl Appelquist, 2013). Coastlines with a free fetch greater than 10 km may also be classed as protected if the local geology or wind and wave climate is such that wave action is limited, this is indicated by the presence of saltmarshes (Mangor, 2004; Rosendahl Appelquist, 2013). Free fetch can easily be determined using most maps and a GIS dataset of saltmarsh extents is available to download (UK Government, 2016).

2.1.2 Tidal classification

The tidal range influences how vulnerable coastlines are to wave energy (McLaughlin and Cooper, 2010) and flooding (Rosendahl Appelquist, 2013). The greater the tidal range the lower the probability that high tide and high waves will coincide, hence the probability of wave related erosion (McLaughlin and Cooper, 2010) and the probability of flooding (Rosendahl Appelquist, 2013) are reduced. In addition, wide intertidal zones in which wave energy can dissipate are often present in areas with high tidal ranges (McLaughlin and Cooper, 2010). The tidal classification, i.e. whether it is macrotidal, mesotidal or microtidal, is considered adequate to assess tidal range as a hazard (Davies and Moses, 1964; Rosendahl Appelquist, 2013) and can be found for all British estuaries in Davidson (1991).

2.1.3 Flooding

Flooding increases the probability of landfills eroding both due to the movement of water over the site (Laner et al., 2008) and because infiltration of high volumes of water can adversely affect the structural integrity of the waste (Blight and Fourie, 2005). In addition, the build-up of water pressure behind a flood defence can cause it to fail, exposing waste (Cooper et al., 2013). The GIS dataset Risk of Flooding from Rivers and Sea (RoFRS) (Environment Agency, 2016b), shows the residual flood zones after mitigation by flood defences broken down into four categories: Very Low (annual probability <0.1%), Low (0.1% ≤ annual probability <1%), Medium (1% ≤ annual probability < 3.3%), and High (3.3% ≤ annual probability) and was used to assess flooding as a driver of erosion/landfill breaching (Environment Agency, 2013a).

2.1.4 Landfill position

The closer the landfill is to mean high water, the greater the risk of it being eroded. There are discrepancies in the position of the high water line between different OS and EA datasets due to different update frequencies and scales used (e.g. Environment Agency, 2016a; Ordnance Survey, 2016). This research used the (mean) High Water line in the OS Boundary-Line dataset (Ordnance Survey, 2016) as it was the most recently updated of the large scale datasets (1:10,000) and OS data are used to produce EA datasets (Environment Agency, 2016a).

2.1.5 Exposed boundary

The length of the landfill boundary exposed to wave impact will also influence the probability of waste being eroded and can be determined by comparing the Historic Landfill Sites National Dataset (Environment Agency, 2017) to the High Water line in the OS Boundary-Line dataset (Ordnance Survey, 2016).

2.1.6 Defence condition and defence type

The likelihood of historic coastal landfills eroding and releasing waste is linked to whether there are effective flood defences present. The probability of flood defences breaching is linked to the probability of them overtopping and coastal erosion, which are already accounted for within the assessment, and their current state of repair and type (Defra and Environment Agency, 2007; Environment Agency, 2010b; Scott Wilson, 2008), which are recorded in the EA's Spatial Flood Defences GIS dataset (UK Government, 2016).

2.1.7 Coastal slope

The shallower the coastal slope (below mean high water) the lower the rate of coastal erosion (Palmer et al., 2011). The Portal for Bathymetry online map (European Marine Observation and Data Network, 2016) depth profile function was used to approximate distances between landfills and the 20 m isobaths as a proxy for coastal slope (after Palmer et al., 2011).

2.1.8 Sediment balance

There is a paucity of national scale erosion and accretion (rate) mapping for England; however, Shoreline Management Plans exist for the entire coast and include data indicating whether areas are eroding or accruing sediment (e.g. Environment Agency, 2010a; Royal Haskoning, 2009). Where the plans provide more than one erosion scenario, e.g. No Active Intervention (NAI) and With Present Management (WPM), WPM data were used to determine the value of the sediment balance parameter as they account for any artificial sediment recharge that may be taking place.

2.1.9 Buffer zone

The presence of vegetated saltmarshes can significantly attenuate the impact of waves upon flood defences, dissipating up to half of the wave energy in the first 10-20 metres of saltmarsh surface, reducing the risk of defences being overtopped or breached (Committee on Climate Change, 2013; Möller and Spencer, 2002). A GIS dataset of saltmarsh extent was used to determine the average width of saltmarsh in front of the landfill (UK Government, 2016).

2.1.10 Landfill volume

Existing landfill risk ranking methods (Alaska Department of Environmental Conservation, 2015; 2009; Laner et al., 2008; Neuhold, 2013; Neuhold and Nachtnebel, 2011) determine the hazard posed by assuming the entire landfill will erode, as saturated waste is known to be mechanically unstable (Blight and Fourie, 2005; Liang et al., 2015). The area of historic landfill sites can be determined from the Historic Landfill Sites National Dataset (Environment Agency, 2017) and GIS mapping software, e.g. ArcMap; however, the dataset does not provide information on waste volumes. Waste volume data for some sites can be obtained from local authorities[i], elsewhere volume can be estimated by comparing historic records of site topography to the present topography or using monitoring well depths (where present) in conjunction with the landfill's area.

However, it seems unlikely that entire landfills would erode as waste is often deposited in discrete cells, where the walls are more resilient to erosion, and breaches in flood defences are likely to be quickly repaired before all of the waste is released. Therefore, to assess the magnitude of the hazard from eroded waste materials, consideration also needs to be given to how quickly waste materials are likely to erode as well as how much waste is present in total. Hence, parameters that are proxies for the erosion rate, i.e. wave energy, tidal classification, landfill position, defence type, coastal

[i] See supplementary information Table S1

slope and buffer zones, are included in the landfill hazard sub-index as well as the coastal drivers and landfill vulnerability sub-indices.

2.1.11 Landfill type

The Historic Landfill Sites National Dataset (Environment Agency, 2017) provides an indication of whether sites contain inert, industrial, commercial, household, special waste, liquids/sludge or if the type of waste is unknown. Just 37% of historic coastal landfill sites contain only a single waste type, 45% of the sites contain a mixture of waste types in unknown proportions and 18% of the sites have no record of the waste received. The range of materials and contaminant concentrations in each waste type vary depending on when waste was deposited (Parfitt, 2009; Quaghebeur et al., 2013), but only 44% of England's historic coastal landfills have both the opening and closing dates recorded.

Even where waste types and operating periods are known, material types and contaminant concentrations are highly variable, e.g. metal concentrations can vary by up to four orders of magnitude between and within sites (Brand and Spencer, in review). Contaminant concentrations, speciation and behaviour are site specific and vary at the micro-scale, and, therefore, representative sampling is challenging and impracticable for a regional or national scale screening assessment (Brand and Spencer, in review; Neuhold, 2013).

The maximum permissible (leachable) concentrations of contaminants in materials being landfilled vary with the landfill site type, e.g. sites that are permitted to take hazardous waste (also known as special waste) are allowed maximum (leachable) concentrations of mercury 200 times higher, and of chromium 140 times higher, than inert sites (Council Decision, 2003). Therefore, for this research the site type is used as a proxy for ranking the severity of the hazard from contaminants in the waste (Alaska Department of Environmental Conservation, 2015; Singh et al., 2009). The severity increases in the order: inert < Municipal Solid Waste (MSW) or commercial < industrial < special waste (Alaska Department of Environmental Conservation, 2015; Council Decision, 2003; NetRegs, n.d.; Singh et al., 2009). Liquids/sludge contain chemical wastes, sewage sludge and industrial wastewater mixed with municipal solid waste (Environment Agency, 2013b), but no information could be found to indicate how hazardous they are in relation to other waste types. However, as historic coastal landfill sites typically pre-date regulations controlling which chemicals are disposed of (Brand et al., 2017), liquids/sludge landfills may contain chemical wastes that would not be accepted at modern landfills for special waste, therefore, in the absence of better data they have been assigned the highest hazard rating. Landfills where the contents are classified as unknown have also been assigned the highest hazard rating as they may contain liquids/sludge.

2.1.12 Salinity

Metal release from waste is significantly higher in saline waters compared to freshwaters (Brand, 2017) and is included as a parameter by using salinity zones determined using the Joint Nature Conservation Council's (JNCC) Variable Salinity Areas dataset (McBreen et al., 2011).

2.1.13 Dissolved contaminants

There are insufficient data to determine the release of contaminants into the water column by leaching (Brand, 2017; Brand et al., 2017). However, it is possible to rank the hazard posed by leached contaminants using the amount of waste eroded and waste type as proxies for the maximum mass of contaminants that could leach, and by considering dilution in the receiving waters. Here, the tidal prism was used as a proxy for the total effective volume of water and was calculated using the average tidal range from Shoreline Management Plans and the estuary's transitional area recorded in the Water Framework Directive (WFD) transitional and coastal waterbodies cycle 2 dataset (UK Government, 2016). For estuaries large enough to be split into multiple transitional zones in the WFD dataset, only the zone adjacent to the landfill was considered. This potentially overestimates the dilution of contaminants for landfill sites that are on tributaries that are not considered independently in the WFD dataset. However, this level of accuracy in determining dilution was considered appropriate given the uncertainty associated with the concentrations of contaminants in the waste and their mobility.

2.1.14 Human impact

In the intertidal zone, humans are most likely to come into contact with any eroded waste or released contaminants during recreational use of beaches. The distances between historic coastal landfills and bathing water catchments shown in the EA areas affecting bathing waters dataset (UK Government, 2016) were used as a proxy for the quantities of solid waste materials and dissolved contaminants that humans may come into contact with. This was based on the assumption that the greater the distance from the source of the waste, the greater the dispersion of the waste and dilution of the contaminants.

2.1.15 Designated sites

There exists a multitude of environmentally designated sites around England. The availability of GIS datasets for those highlighted as being vulnerable to contaminants from historic coastal landfills by Cooper *et al* (2013), and others that fall within the coastal (flood) zone are shown in Supplementary information Table S2. For the purposes of this assessment heritage coasts were also treated as designated sites. Designated sites upstream as well as downstream of the landfills were included to account for tidal movement of contaminants.

2.1.16 Seafood

Seaweed, crustaceans, other shellfish and fish may be harvested from the intertidal zone and tidal waters for human consumption. Only GIS datasets relating to shellfish waters were available for the assessment: Cefas's Classified Bivalve Mollusc Harvesting Areas GIS dataset (O. Morgan, pers. comm., email, 2/11/2015) and the Shellfish Waters GIS dataset (Defra, 2016). Similar to assessing the vulnerability of human receptors, distances between these areas and historic coastal landfill sites were used as a proxy for the quantities of solid waste materials and dissolved contaminants that may reach these areas.

2.2 Calculation of the sub-indices, waste release and pollution indices and overall risk index

A summation method was used to combine the severity scores to determine the values of the sub-indices (1) (Khouakhi et al., 2013; Musekiwa et al., 2015; Ramieri et al., 2011). No weightings were directly applied to individual parameters within the sub-indices. Where a different number of parameters are used for each of the sub-indices, normalising each sub-index value to a percentage allows them to be combined into the overall risk index without any one sub-index dominating the overall risk score (McLaughlin and Cooper, 2010). Therefore, the four sub-indices were normalised to percentages using (2) and values from Table 2, before being combined into the waste release index and pollution index using (3) and (4) respectively. The overall risk index was then calculated using (5). All three indices have value ranges from 0 to 100.

(1) Calculation of the sub-indices

$$Sub\text{-}index = \sum severity\ scores$$

(2) Calculation of the normalised sub-indices (after McLaughlin and Cooper, 2010)

$$Normalised\ sub\text{-}index = \frac{\sum severity\ scores - min.possible\ score}{max.possible\ score - min.possible\ score}\ x\ 100$$

Table 2: Minimum and maximum possible sub-indices scores (before normalisation)

Sub-index	Minimum possible score	Maximum possible score
coastal drivers	3	15
landfill vulnerability	7	35
landfill hazard	10	50
environmental vulnerability	3	15

(3) Calculation of the waste release index

$$Waste\ release\ index = \frac{normalised\ coastal\ drivers + normalised\ landfill\ vulnerability}{2}$$

(4) Calculation of the pollution index

$$Pollution\ index = \frac{normalised\ landfill\ hazard + normalised\ environmental\ vulnerability}{2}$$

(5) Calculation of the overall risk index

$$Overall\ risk\ index = \frac{waste\ release\ index + pollution\ index}{2}$$

3. Testing the risk screening assessment methodology

3.1 Study site selection

Eight historic coastal landfills were selected for testing the screening assessment methodology (Table 3 and Figure 2). The landfills are distributed over four estuaries in southeast England, but some are adjacent to each other, which allows testing of the method for sensitivity to changes in factors such as the distance between the landfill and mean high water. All of the landfills were chosen from the same region as the method needs to distinguish risk at a local level in the case that remediation funds are allocated locally. As adjacent sites could be affected by the same extreme event, Martins Farm North and Martins Farm South were used to test whether the risk ranking would be affected if sites in close proximity were subject to a joint assessment as well as individual assessments. To test the effect of giving individual weightings to parameters, the analysis was done twice, once with the default method given and once with double weighting applied to the unique landfill hazard parameters (landfill volume, landfill type, salinity and dissolved contaminant), i.e. those parameter scores were multiplied by two, and the landfill hazard normalisation calculation adjusted accordingly by using a minimum possible score = 14 and a maximum possible score = 70.

Table 3: Screening assessment test site histories

Name and landfill database reference no.[1]	Operating period[2]	Type[3]	Volume[2] (m^3)	Flood defences[1]
Common Road EAHLD01226	1970-1993	Household, commercial and industrial	450,000	Partly defended
Hadleigh Marsh EAHLD01181	1980-1987	Household and commercial	500,000	Landfill is the flood defence
Leigh Marshes EAHLD00531	1955-1967	Household, commercial and industrial	800,000	Yes
Martins Farm North EAHLD01246	1960-1995	Household, commercial and industrial	1,400,000	Yes
Martins Farm South EAHLD01241	1985-1995	Household, commercial and industrial	1,200,000	Yes
Newlands EAHLD01178	1954-1989	Household, commercial and industrial	1,000,000	Landfill is the flood defence
Park Drive EAHLD01739	1974-1994	Household, commercial and industrial	800,000	Landfill is the flood defence
Sea Wall EAHLD01228	1988-1991	Household, commercial and industrial	275,000	Landfill is the flood defence

[1] GIS datasets from UK Government (2016)
[2] Site records (A. Brown, pers. comm., email, 26/10/2015), except Leigh Marshes operating period from Environment Agency (2013b) and volume estimated using GIS data from UK Government (2016) and trial pit depths recorded in a report by Halcrow (2012).
[3] Environment Agency (2013b)

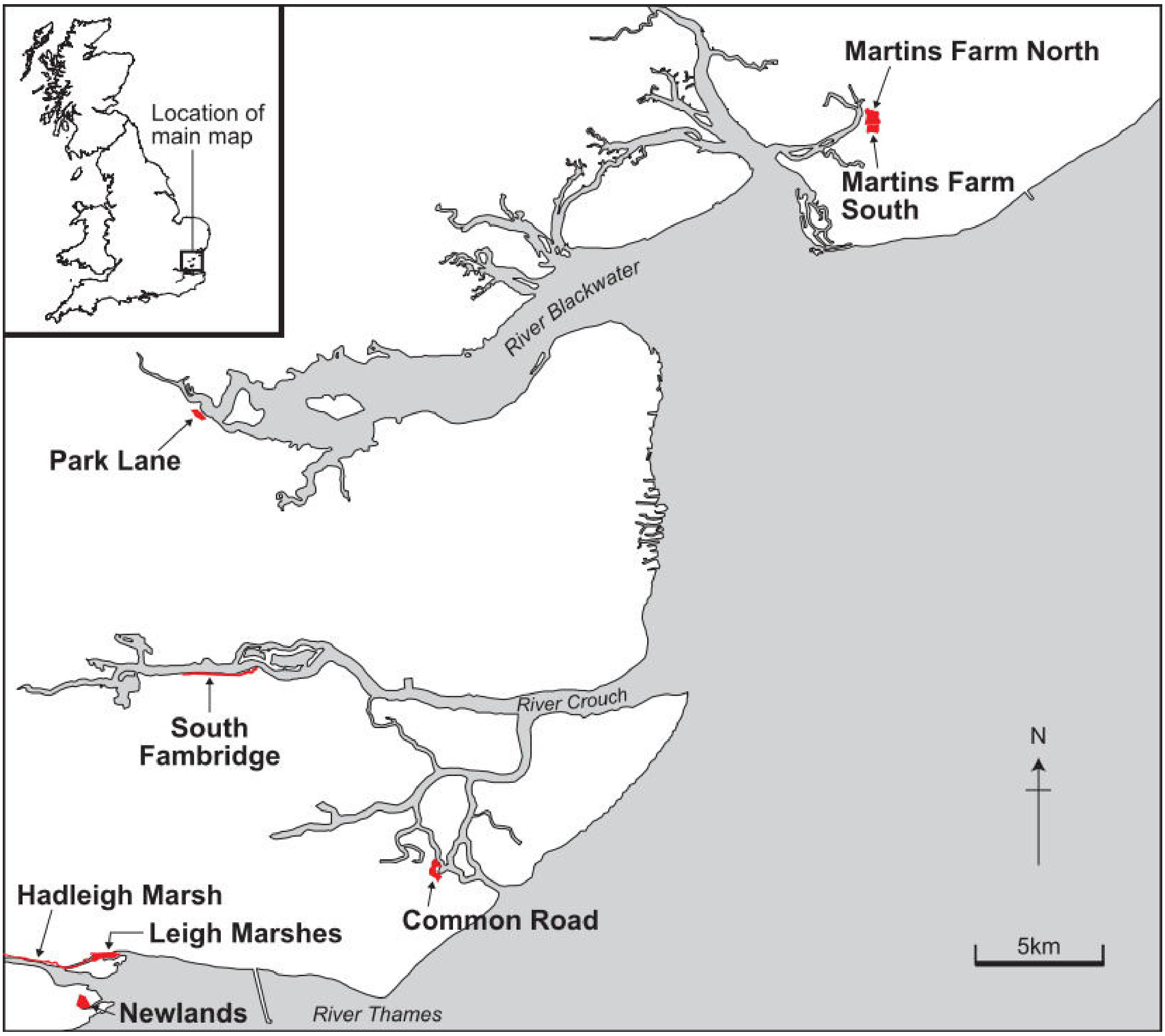

Figure 2: Map showing the locations of the 8 historic coastal landfills selected for testing the risk screening assessment (created using data © Environment Agency copyright and/or database right 2017. All rights reserved. Contains information © Local Authorities. © Crown copyright and database rights 2004 Ordnance Survey 100024198)

3.2 Results

The parameter severity scores, sub-indices values and indices values are shown in Table 4 for the default methodology with no individual parameter weightings applied. Of the sites tested, Hadleigh Marsh had the highest overall risk index, with a value of 53.5, and Martins Farm South had the lowest overall risk index, with a value of 40.9. The results of the analysis with double weighting applied to the unique landfill hazard parameters are shown in Table 5, the combined Martins Farm sites moved one place higher and Martins Farm North moved two places higher in the ranking under this methodology.

Table 4: Results of assessing the test sites, sites shown left to right from highest to lowest overall risk index value - no weightings given to individual paramters

Parameter being scored	**Hadleigh Marsh**	**Sea Wall**	**Common Road**	**Martins Farm- both combined**	**Leigh Marshes**	**Newlands**	**Martins Farm North**	**Park Drive**	**Martins Farm South**
Wave energy	1	1	1	1	1	1	1	1	1
Tidal classification	1	1	1	1	1	1	1	1	1
Flooding	2	3	1	1	3	1	1	1	1
Landfill position	5	5	5	4	5	5	4	4	1
Exposed boundary length	5	5	3	2	3	3	2	1	1
Defence condition	3	3	3	4	3	3	4	2	3
Defence type	5	5	4	3	2	5	3	5	3
Coastal slope	4	1	1	1	4	5	1	5	1
Sediment balance	5	5	3	5	3	5	5	3	5
Buffer zone	4	5	5	5	5	1	5	4	5
Landfill volume	1	1	1	5	2	2	3	2	3
Landfill type	2	3	3	3	3	3	3	3	3
Salinity	4	5	5	3	4	4	3	4	3
Dissolved contaminant	1	2	2	4	1	1	4	2	4
Human impact	5	1	5	5	5	5	5	5	5
Designated sites	5	5	5	5	5	5	5	5	5
Seafood	2	5	5	4	2	3	4	1	3
Coastal drivers sub-index	4	5	3	3	5	3	3	3	3
Landfill vulnerability sub-index	31	29	24	24	25	27	24	24	19
Landfill hazard sub-index	28	29	28	30	28	28	28	31	25
Environmental vulnerability sub-index	12	11	15	14	12	13	14	11	13
Normalised coastal drivers sub-index	8.3	16.7	0	0	16.7	0	0	0	0
Normalised landfill vulnerability sub-index	85.7	78.6	60.7	60.7	64.3	71.4	60.7	60.7	42.9
Normalised landfill hazard sub-index	45.0	47.5	45.0	50.0	45.0	45.0	45.0	52.5	37.5
Normalised environmental vulnerability sub-index	75.0	66.7	100	91.7	75.0	83.3	91.7	66.7	83.3
Waste release index	47.0	47.6	30.4	30.4	40.5	35.7	30.4	30.4	21.4
Pollution index	60.0	57.1	72.5	70.8	60.0	64.2	68.3	59.6	60.4
Overall risk index	**53.5**	**52.4**	**51.4**	**50.6**	**50.2**	**49.9**	**49.3**	**45.0**	**40.9**

Table 5: Results of assessing the test sites, sites shown left to right from highest to lowest overall risk index value - unique landfill hazard parameters double weighted

Parameter being scored	Hadleigh Marsh	Sea wall	Martins Farm-both sites	Common Road	Martins Farm North	Leigh Marshes	Newlands	Park Drive	Martins Farm South
Coastal drivers sub-index	4	5	3	3	3	5	3	3	3
Landfill vulnerability sub-index	31	29	24	24	24	25	27	24	19
Landfill hazard sub-index	36	40	45	39	41	38	38	42	38
Environmental vulnerability sub-index	12	11	14	15	14	12	13	11	13
Normalised coastal drivers sub-index	8.3	16.7	0	0	0	16.7	0	0	0
Normalised landfill vulnerability sub-index	85.7	78.6	60.7	60.7	60.7	64.3	71.4	60.7	42.9
Normalised landfill hazard sub-index	39.3	46.4	55.4	44.6	48.2	42.9	42.9	50.0	42.9
Normalised environmental vulnerability sub-index	75.0	66.7	91.7	100.0	91.7	75.0	83.3	66.7	83.3
Waste release index	47.0	47.6	30.4	30.4	30.4	40.5	35.7	30.4	21.4
Pollution index	57.1	56.5	73.5	72.3	69.9	58.9	63.1	58.3	63.1
Overall risk index	**52.1**	**52.1**	**51.9**	**51.3**	**50.1**	**49.7**	**49.4**	**44.3**	**42.3**

4. Discussion

The coastal drivers sub-index ranked Leigh Marshes and Sea Wall in South Fambridge as the sites potentially subjected to the greatest drivers of erosion, followed by Hadleigh Marsh. However, the landfill vulnerability sub-index indicated that Leigh Marshes is better protected from the coastal drivers than Sea Wall and Hadleigh Marsh, which reflects the fact that it has a much shorter length of boundary facing mean high water and is separated from the estuary by a flood defence. In contrast, Sea Wall and Hadleigh Marsh are both waste-filled flood embankments with several kilometres of exposed boundary. The waste release index, which combines the coastal drivers and landfill vulnerability sub-indices, indicated the two flood embankments (Sea Wall in South Fambridge and Hadleigh Marsh) are the two most likely test sites to release solid waste to the environment, reflecting their exposure to their estuaries, having no flood defences separating them from the water and having very long boundaries adjacent to mean high water, increasing the probability that at least part of the landfill sites will breach.

In contrast, the two waste-filled flood embankments were ranked low in the range of pollution index values suggesting that, if waste erodes from them, they are likely to cause comparatively less pollution than the other sites tested. This reflects the relatively small volumes of waste in the two flood embankments, combined with the high levels of dilution at Hadleigh Marsh landfill site and the absence of bathing water catchments in the estuary at the Sea Wall in South Fambridge landfill site. However, the two waste-filled flood embankments had the two highest overall risk index values reflecting that, for the test sites, the range of waste release index values (range = 26.2) is greater than the range of pollution index values (range = 15.4) and therefore the waste release index has greater influence in determining the overall risk index ranking of the test sites. The limited range of pollution index values reflects the very similar waste contents and ecological environments of the eight sites, and the greater range of waste release index values reflects the greater range of vulnerabilities of the landfill sites to coastal drivers, particularly differences in defences and the lengths of their boundaries.

The inclusion of some parameters within more than one sub-index means that the sub-indices are not fully independent of each other and the duplicated parameters have greater influence upon the overall risk index[ii]. In addition, the greater number of parameters in the landfill vulnerability sub-index compared to the coastal drivers sub-index means the waste release index and overall risk index are more sensitive to changes in the coastal drivers sub-index parameters than the landfill vulnerability sub-index parameters. Similarly, the greater number of parameters in the landfill hazard sub-index compared to the environmental vulnerability sub-index means the pollution index and overall risk index are more sensitive to changes in the environmental vulnerability sub-index parameters than the landfill hazard sub-index parameters. It could be argued that the vulnerability of receptors is more important in determining pollution risk than the chemical content of the material released from an eroding landfill site as waste material has the potential to physically and chemically alter the coastal or estuarine environment if eroded, but studies of the impact of landfill debris on the marine environment are limited (Pope et al., 2011). These potential issues highlight that further consideration is needed to determine which parameters, if any, are in reality more significant in determining the overall risk and, hence, whether weightings should be applied to increase their influence on the final risk rankings. To demonstrate the importance of weighting individual parameters, testing the application of a double weighting to the unique landfill hazard parameters increased the risk ranking of the combined Martins Farm sites one place and Martins Farm North two places. Weightings must be specific to the combination of parameters and indices being used; therefore, to determine weightings with any useful level of accuracy would require input from experts in coastal processes, landfill engineering stability and contamination, and ecology. In addition to weighting of parameters, it may also be appropriate to include a distinction between different types of ecological sites to ensure that those most difficult to replace or rehabilitate are given priority when considering which landfill sites to remediate first. The necessary consultations were beyond the scope of this research, but should be considered in any future developments of the risk screening assessment. The consultations should also consider whether sites in close proximity should be subject to a joint assessment as well as individual assessments in case there is an event of sufficient magnitude to breach multiple sites, e.g. a storm surge, and, if so, what the minimum separation between sites should be before they are only assessed independently. To demonstrate the importance of this, considering Martins Farm North and Martins Farm South in combination ranked them 4th by overall risk index compared to 6th and 8th when only individual sites were considered.

[ii] See Supplementary information - Sensitivity analysis of the risk screening assessment

The value of the overall risk index can range from 0 to 100 under the proposed scoring system. As there are over 1200 (currently known) historic coastal landfills to be ranked (Brand et al., 2017), there will be multiple landfill sites with similar overall risk index values. If a series of overall risk index value thresholds were set to provide categories of risk, e.g. very high, high, moderate, low and very low, then this would mitigate the issue of having multiple sites with the same or similar index values. Note a zero risk category is deliberately not included as there is always a residual risk of a site eroding and causing pollution (Neuhold and Nachtnebel, 2011). A categorical risk approach would also have the advantage of allowing the end-user greater discretion in determining the order in which sites are considered for further investigation and/or remedial action, which would better support management of limited budgets. For example, if all sites in a risk category are given the same priority for remediation, rather than using the overall risk score to rank them individually, it would allow multiple sites with low remediation costs to be addressed instead of a single site within the same category that has a higher overall risk score and a higher remediation cost. However, such categories cannot be implemented until a much greater number of sites have been assessed to provide a benchmark of the levels at which such fixed thresholds should be set.

To undertake a national scale risk screening assessment using this methodology would be relatively straightforward. The majority of parameter scores can be determined from data tables in literature, or from GIS datasets either by reading data tables, creating buffer zones or directly measuring distances. Only the sediment balance and dissolved contaminant scores require searching documents, e.g. Shoreline Management Plans (SMPs), for data. Based on the test it is estimated that it would take approximately 3 months to assess the circa 1200 sites around the coast of England, but this timescale could be significantly reduced using efficiencies in data collection, e.g. tidal prism would not need to be calculated for each individual site and some scores could be automatically calculated in GIS software, e.g. landfill volume. If the assessment were divided by SMP area then it would take less than one week per SMP (before efficiencies) and could easily be integrated into one of the periodic SMP updates.

5. Conclusion

A new risk screening assessment method has been proposed that can support coastal managers in identifying which historic coastal landfill sites pose the greatest pollution risk at a national scale for minimal cost using existing datasets. The highest risk sites can then be prioritised for further investigation, including ground-truthing, or remedial works as appropriate. The risk screening assessment provides a snapshot of the current highest risk sites and should be updated as the underlying datasets are modified to reflect changes to factors such as site condition, e.g. due to maintenance works, or flood extent, e.g. due to climate change related sea level rise or changes to defences.

Prior to the assessment being implemented nationally, consultations should be carried out with experts in coastal processes, landfill engineering stability and contamination, and ecology to ensure the parameter severity scores, sub-indices and indices calculations are appropriately weighted to reflect their contribution to the overall risk of historic coastal landfill sites eroding and causing pollution. These also need to be agreed with appropriate regulators. Currently parameters representing the total landfill volume and contaminant concentrations in the waste have the lowest influence on the overall risk score, and parameters representing the probability of waste being released, the rate at which it will be released, and the vulnerability of receptors are of much greater importance in determining the overall risk score. This suggests the uncertainty and incompleteness of the data representing the landfill volumes and contaminant concentrations in waste are not a major obstacle to assessing the risk of pollution from historic coastal landfill sites, and that resources should not be expended on attempting to improve the accuracy of these parameter datasets, particularly given the difficulties of obtaining representative contaminant data and the high costs involved (Brand and Spencer, in review). However, the importance of the landfill volume and contaminant concentrations in the waste in determining the overall risk score may increase once weightings have been added to the risk screening assessment parameters and indices.

Testing the risk screening assessment, by applying it to eight historic coastal landfills in southeast England, found that despite their relatively small volumes, the only two waste-filled flood embankments screened (Hadleigh Marsh and Sea Wall in South Fambridge) pose the greatest overall risk of pollution. This is due to their relatively high exposure to drivers of coastal erosion and vulnerability to erosion, which means they are more likely to breach than the other sites screened and, if breached, are likely to release waste at a greater rate than most other sites screened. This means that these two sites should be given priority for expenditure on further investigation and/or remedial actions ahead of the other six sites screened.

Acknowledgements

Funding: This work was carried out as part of a project funded by the Environment Agency and Southend Borough Council, and supported by Essex County Council.

References

Alaska Department of Environmental Conservation, 2015. Waste Erosion Assessment and Review (WEAR), Available online at http://dec.alaska.gov/eh/sw/wear.html [Accessed 12th January 2016], p. 43.

Blight, G., Fourie, A., 2005. Catastrophe revisited–disastrous flow failures of mine and municipal solid waste. Geotechnical & Geological Engineering 23, 219-248.

Brand, J.H., 2017. *Assessing the risk of pollution from historic coastal landfills.* PhD, Queen Mary University of London.

Brand, J.H., Spencer, K.L., in review. Potential contamination of the coastal zone by eroding historic landfills.

Brand, J.H., Spencer, K.L., O'Shea, F.T., Lindsay, J.E., 2017. Potential pollution risks of historic landfills on low-lying coasts and estuaries. WIREs Water 2017, e1264. doi: 10.1002/wat2.1264, 12.

Committee on Climate Change, 2013. *Managing the land in a changing climate - Adaptation sub-committee progress report 2013* [Online]. Available: https://www.theccc.org.uk/publication/managing-the-land-in-a-changing-climate/ [Accessed 21st April 2016].

Cooper, N., Bower, G., Tyson, R., Flikweert, J., Rayner, S., Hallas, A., 2013. Guidance on the management of landfill sites and land contamination on eroding or low-lying coastlines, CIRIA, C718 (ISBN: 978-0-86017-721-0). www.ciria.org, London.

Council Decision, 2003. Council decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC (2003/33/EC), Official Journal of the European Communities.

Davidson, N.C., 1991. Nature conservation and estuaries in Great Britain. JNCC, UK.

Davies, J.L., Moses, C.A., 1964. A Morphogenic Approach to World Shorelines. Zeitschrift für Geomorphologie 8, 27-42.

Defra, 2016. *MAGIC Dataset listings and download options* [Online]. Available: http://magic.defra.gov.uk/Dataset_Download_Summary.htm [Accessed 12th May 2016].

Defra, Environment Agency, 2007. Performance and Reliability of Flood and Coastal Defences, in: Bujis, F., Simm, J., Wallis, M., Sayers, P. (Eds.), London, UK.

Denner, K., Phillips, M.R., Jenkins, R.E., Thomas, T., 2015. A coastal vulnerability and environmental risk assessment of Loughor Estuary, South Wales. Ocean & Coastal Management 116, 478-490.

Environment Agency, 2010a. Essex and South Suffolk Shoreline Management Plan 2, United Kingdom.

Environment Agency, 2010b. Northern Area Tidal Modelling. Volume 4: Breach flood mapping, in: Mott MacDonald (Ed.). Environment Agency, Lincoln.

Environment Agency, 2013a. *What's in your backyard? Flood maps.* [Online]. Available: http://www.environment-agency.gov.uk/homeandleisure/37837.aspx [Accessed 10th May 2013].

Environment Agency, 2013b. *What's in your backyard? Historic landfill* [Online]. Environment Agency. Available: http://www.environment-agency.gov.uk/homeandleisure/37829.aspx [Accessed 5th June 2013].

Environment Agency, 2016a. *WFD Transitional and Coastal Waterbodies Cycle 2* [Online]. Environment Agency,. Available: https://data.gov.uk/dataset/wfd-transitional-and-coastal-waterbodies-cycle-2 [Accessed 11th May 2016].

Environment Agency, 2016b. *What's in your backyard? Risk of flooding from rivers and sea.* [Online]. Available: http://watermaps.environment-agency.gov.uk/wiyby/wiyby.aspx?&topic=floodmap#x=581028&y=185284&scale=9 [Accessed 18th February 2016].

Environment Agency, 2017. Historic Landfills GIS Shapefile (downloaded 13/4/2017).

European Marine Observation and Data Network, 2016. *Portal for Bathymetry* [Online]. Available: http://portal.emodnet-bathymetry.eu/mean-depth-full-coverage [Accessed 31st May 2016].

Gill, J., Ainee, A.M., Anwar, S., Omar, K., 2014. Towards the implementation of continuous coastal vulnerability index in Malaysia: a review. Jurnal Teknologi 71, 1-10.

Gormley, A., Pollard, S., Rocks, S., 2011. Guidelines for environmental risk assessment and management - Green Leaves III, November 2011 ed. Defra, UK.

Halcrow Group Ltd, 2012. Contamination Risk Assessment Report: Two Tree Island and Hadleigh Marshes Contamination Study, Swindon, UK.

Khouakhi, A., Snoussi, M., Niazi, S., Raji, O., 2013. Vulnerability assessment of Al Hoceima bay (Moroccan Mediterranean coast): a coastal management tool to reduce potential impacts of sea-level rise and storm surges. Journal of Coastal Research Proceedings 12th International Coastal Symposium (Plymouth, England), Special Issue No. 65, 968-973.

Kumar, D., Alappat, B., 2005. Evaluating leachate contamination potential of landfill sites using leachate pollution index. Clean Techn Environ Policy 7, 190-197.

Kumar, T.S., Mahendra, R.S., Nayak, S., Radhakrishnan, K., Sahu, K., 2010. Coastal vulnerability assessment for Orissa State, east coast of India. Journal of Coastal Research 26, 523-534.

Laner, D., Fellner, J., Brunner, P.H., 2009. Flooding of municipal solid waste landfills - An environmental hazard? Science of the Total Environment 407, 3674-3680.

Laner, D., Fellner, J., Brunner, P.H., Neuhold, C., Kolesar, C., 2008. Environmental relevance of flooded MSW landfills in Austria, ISWA/WMRAS, ISWA/WMRAS World Congress. 3-6, pp. 3-6.

Liang, L., Kuang, S.L., Diao, Z., 2015. Stress and Strain Analysis of Waste Dam and Stability Analysis of Landfill. Applied Mechanics and Materials 744-746, 702-705.

Mangor, K., 2004. Shoreline management guidelines, 3rd ed. ed. DHI Water & Environment, Horsholm, Denmark.

McBreen, F., Askew, N., Cameron, A., Connor, D., Ellwood, H., Carter, A., 2011. *Predictive mapping of seabed habitats in UK waters* [Online]. JNCC. Available: http://jncc.defra.gov.uk/page-5534 [Accessed 2nd August 2016].

McLaughlin, S., Cooper, J., 2010. A multi-scale coastal vulnerability index: a tool for coastal managers? Environmental Hazards 9, 233-248.

Met Office, 2016. *UK regional climates* [Online]. Met Office. Available: http://www.metoffice.gov.uk/climate/uk/regional-climates/ [Accessed 2nd August 2016].

Möller, I., Spencer, T., 2002. Wave dissipation over macro-tidal saltmarshes: Effects of marsh edge typology and vegetation change. Journal of Coastal Research 36, 506-521.

Musekiwa, C., Cawthra, H., Unterner, M., van Zyl, F., 2015. An assessment of coastal vulnerability for the South African coast. South African Journal of Geomatics 4, 123-137.

NetRegs, n.d. *Managing your hazardous or special waste* [Online]. Available: http://www.netregs.org.uk/library_of_topics/waste/hazardous,_special_waste/whats_hazardousspecial_waste.aspx [Accessed 15th April 2016].

Neuhold, C., 2013. Identifying flood-prone landfills at different spatial scales. Natural Hazards 65, 2015-2030.

Neuhold, C., Nachtnebel, H.P., 2011. Assessing flood risk associated with waste disposals: methodology, application and uncertainties. Natural Hazards 56, 359-370.

Okaneya, M., Laner, D., Stanisavljevic, N., Iino, F., Oki, T., 2013. Prioritization of landfills subject to flood risk in Serbia, 6th PSU-UNS International Conference on Engineering and Technology (ICET-2013). PSU-UNS. 1-3, Novi Sad, Serbia, pp. 1-3.

Ordnance Survey, 2016. *OS OpenData* [Online]. Available: https://www.ordnancesurvey.co.uk/opendatadownload/products.html [Accessed 10th May 2016].
Palmer, B., Van der Elst, R., Mackay, F., Mather, A., Smith, A., Bundy, S., Thackeray, Z., Leuci, R., Parak, O., 2011. Preliminary coastal vulnerability assessment for KwaZulu-Natal, South Africa, ICS. Journal of Coastal Research. 1390-1395, Poland, pp. 1390-1395.
Parfitt, J., 2009. Taking out the rubbish: Municipal waste composition, trends and futures. Resource Futures, Bristol, UK.
Pope, N.D., O'Hara, S.C., Imamura, M., Hutchinson, T.H., Langston, W.J., 2011. Influence of a collapsed coastal landfill on metal levels in sediments and biota--a portent for the future? Journal of Environmental Monitoring 13, 1961-1974.
Quaghebeur, M., Laenen, B., Geysen, D., Nielsen, P., Pontikes, Y., Van Gerven, T., Spooren, J., 2013. Characterization of landfilled materials: screening of the enhanced landfill mining potential. Journal of Cleaner Production 55, 72-83.
Ramieri, E., Hartley, A., Barbanti, A., Santos, F., Gomes, A., Hilden, M., Laihonen, P., Marinova, N., Santini, M., 2011. Methods for assessing coastal vulnerability to climate change. ETC CCA Technical Paper 1/2011. European Environment Agency. 1-93, Bologna, Italy, pp. 1-93.
Rosendahl Appelquist, L., 2013. Generic framework for meso-scale assessment of climate change hazards in coastal environments. J Coast Conserv 17, 59-74.
Rosendahl Appelquist, L., Balstrøm, T., 2014. Application of the Coastal Hazard Wheel methodology for coastal multi-hazard assessment and management in the state of Djibouti. Climate Risk Management 3, 79-95.
Royal Haskoning, 2009. Port of Falmouth Development Initiative - Final Report, United Kingdom.
Rygel, L., O'sullivan, D., Yarnal, B., 2006. A Method for Constructing a Social Vulnerability Index: An Application to Hurricane Storm Surges in a Developed Country. Mitigation and Adaptation Strategies for Global Change 11, 741-764.
Sayers, P.B., Gouldby, B.P., Simm, J.D., Meadowcroft, I., Hall, J., 2003. Risk, performance and uncertainty in flood and coastal defence - a review, May 2003 ed. Defra/Environment Agency, UK, pp. 1-141.
Scott Wilson, 2008. London Boroughs of Wandsworth, Merton, Sutton and Croydon. Level 1 Strategic Flood Risk Assessment. Appendix C - Breach Modelling and Riverside Topographic Analysis, London.
Sharma, A., Meesa, S., Pant, S., Alappat, B.J., Kumar, D., 2008. Formulation of a landfill pollution potential index to compare pollution potential of uncontrolled landfills. Waste Management & Research 26, 474-483.
Singh, R.K., Datta, M., Nema, A.K., 2009. A new system for groundwater contamination hazard rating of landfills. Journal of Environmental Management 91, 344-357.
UK Government, 2016. *Partner Data Catalogue* [Online]. Available: http://environment.data.gov.uk/ds/partners/#/partners/partners [Accessed 12th April 2016].
Wamsley, T., 2015. Guidance for Developing Coastal Vulnerability Metrics. Journal of Coastal Research 31, 1521-1530.
Weber, R., Watson, A., Forter, M., Oliaei, F., 2011. Review Article: Persistent organic pollutants and landfills - a review of past experiences and future challenges. Waste Management & Research 29, 107-121.

Risk screening assessment for ranking historic coastal landfills by pollution risk - Supplementary Information

1. Data held by coastal county councils about historic coastal landfill sites

Table S1: Data held by coastal county councils about historic coastal landfill sites around the coast of England. Other coastal county councils only provided location data or did not respond

	County Council holds records relating to the parameter?							
Landfill parameter	**Devon**	**Essex**	**Hampshire**	**Kent**	**Northumberland**	**North Yorkshire**	**Somerset**	**Suffolk**
Operating dates	Yes	Yes	No	Yes	Yes	Yes	No	Yes
Site type	Yes	Yes	No	Yes	Yes	Yes	No	Yes
Site area	Yes	Yes	No	Yes	No	Yes	No	No
Average waste depth	No	Yes	No	No	No	Yes	No	Yes
Waste mass or volume	Yes	Yes	No	No	Yes	Yes	No	Yes
Environmental controls, e.g. leachate management	Yes	Yes	No	Yes	Yes	Yes	No	Yes
Environmental monitoring	Yes	Yes	No	Yes	Yes	Yes	Yes	No response
References (all pers. comm. by email)	S. Price, 18/12/2013	A. Brown, 26/10/2015	D. Emmett, 6/1/2014. A. Galea, 8/1/12014	L. Pronger, 23/12/2013	S. Wardle, 17/12/2013	A. Surman, 20/1/2014	D. Oaten, 6/1/2014	Anon. Suffolk County Council, 8/1/2014

2. Datasets for designated environmental sites and other ecological sites

Table S2: Available datasets for designated environmental sites and other ecological sites

Designated environmental sites	GIS dataset source
Ecology related designated sites	
Environmentally Sensitive Areas (ESAs)	Natural England (2016)
Important Bird Areas (IBAs)	RSPB (2016)
Local Nature Reserves (LNRs)	Natural England (2016)
Marine Protected Areas (MPAs) includes SACs with Marine Components, SPAs with Marine Components, Marine Conservation Zones, Nature Conservation Marine Protected Areas (Scotland only), Marine Nature Reserves (Isle of Man only) and OSPAR MPAs	JNCC (2016)
National Nature Reserves (NNRs)	Natural England (2016)
Priority Habitat Inventory	Natural England (2016)
Recommended Marine Conservation Zones (rMCZs)	Natural England (2016)
RAMSAR	JNCC (2016)
RSPB Reserves	RSPB (2016)
SAC (including SACs with Marine Components)	JNCC (2016)
SPA (including SPAs with Marine Components)	JNCC (2016)
SSSI	Natural England (2016)
Other designated environmental sites	
Areas of Outstanding Natural Beauty (AONB)	Natural England (2016)
Country Park	Natural England (2016)
National Parks	Natural England (2016)
Other environmental sites	
Heritage Coasts	Natural England (2016)

NB All Natural England datasets are currently being moved as part of Defra's Open Data Programme and in the future will be downloadable from environment.data.gov.uk (UK Government, 2016)

3. Sensitivity analysis of the risk screening assessment

A range sensitivity method was applied to the risk screening assessment to determine which parameters have the greatest influence on the indices' values (United States Environmental Protection Agency, 2001). The sensitivity ratio was determined by varying the severity score for each parameter in turn from the baseline value of 3, the mid-point on the severity scale, to the maximum possible severity score of 5 and substituting the resulting index values into (6) (Saltelli et al., 2008; United States Environmental Protection Agency, 2001). Sensitivity ratios were determined for the waste release index, pollution index and overall risk index, the results can be seen in Table S3. The overall risk index is most sensitive to variations in the wave energy and tidal classification severity scores and least sensitive to changes in the landfill volume, landfill type, salinity and dissolved contaminant severity scores.

(6) Calculation of the sensitivity ratio (after United States Environmental Protection Agency, 2001)

$$Sensitivity\ ratio = \frac{\frac{Y_2 - Y_1}{Y_1}\ x\ 100\%}{\frac{X_2 - X_1}{X_1}\ x\ 100\%}$$

Where:
Y_1 = the baseline index value (i.e. when all severity scores = 3)
Y_2 = the index value when the parameter's severity score is X_2
X_1 = the baseline severity score for the parameter being tested
X_2 = the maximum possible severity score for the parameter being tested

Table S3: Results of the Range Sensitivity Ratio analyses (from highest to lowest sensitivity ratio for overall risk)

	Range Sensitivity Ratio for the:		
Parameter	**Waste release index**	**Pollution index**	**Overall risk index**
Wave energy	0.250	0.075	0.163
Tidal classification	0.250	0.075	0.163
Flooding	0.250	0.000	0.125
Human impact	0.000	0.250	0.125
Designated sites	0.000	0.250	0.125
Seafood	0.000	0.250	0.125
Landfill position	0.107	0.075	0.091
Defence type	0.107	0.075	0.091
Coastal slope	0.107	0.075	0.091
Buffer zone	0.107	0.075	0.091
Exposed boundary length	0.107	0.000	0.054
Defence condition	0.107	0.000	0.054
Sediment balance	0.107	0.000	0.054
Landfill volume	0.000	0.075	0.038
Landfill type	0.000	0.075	0.038
Salinity	0.000	0.075	0.038
Dissolved contaminant	0.000	0.075	0.038

References

JNCC, 2016. *Download GIS data - UK-wide digital GIS datasets for internationally designated sites* [Online]. Available: http://jncc.defra.gov.uk/protectedsites/SACselection/gis_data/terms_conditions.asp [Accessed 12th May 2016].

Natural England, 2016. *Natural England - Publications and Maps* [Online]. Available: http://www.gis.naturalengland.org.uk/pubs/gis/GIS_register.asp [Accessed 12th May 2016].

RSPB, 2016. *Mapping and GIS document library* [Online]. Available: http://www.rspb.org.uk/forprofessionals/gis/library.aspx [Accessed 11th May 2016].

UK Government, 2016. *Partner Data Catalogue* [Online]. Available: http://environment.data.gov.uk/ds/partners/#/partners/partners [Accessed 12th April 2016].